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Habitat mosaic of gravel pit as a potential refuge for carabids: a case study from Central Europe

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Abstract: In gravel pits, a mosaic of habitats with various environmental conditions created during mining has a great potential for persistence of many species. We focused on such a mosaic in a gravel pit surrounded by agricultural landscape. We investigated which habitats within sludge deposits in different successional stages (from bare sands to secondary forest) and agriculturally reclaimed area enhanced diversity, species richness and abundance of carabids and supported occurrence of threatened species. Since some of these habitats were extensively managed while others were invaded by the alien plant *Solidago gigantea*, we also tested the effect of management and the cover of *S. gigantea* on carabid assemblages. We found a gradient in carabid assemblages from psammophilous ones in bare sandy soils towards similar assemblages in plots with well-developed vegetation cover. Here, carabid assemblages were represented predominantly by common species of agricultural and forest lands without higher habitat requirements. Contrarily, plots with bare sand could serve as a refuge for rare psammophilous carabid species, which cannot occur in surrounding landscape due to vanished suitable habitats. Therefore, keeping some of habitats in early plant successional states is important for maintaining habitat mosaic and for persistence of such species as well. Management of grasslands and cover of *S. gigantea* had no effect on carabid assemblage. We presume that carabids were likely more affected by vegetation structure and density than species composition.

Nomenclature: Hůrka (1996) for carabids, Danihelka et al. (2012) for plants.

Abbreviations: AICc – Akaike Information Criterion; GLMM – Generalized Linear Mixed Model; NMDS – Non-metric Multidimensional Scaling; R – Reclaimed area; SD – Sludge Deposit.

Introduction

In quarries, sand and gravel pits, quarrying of raw materials inherently constitutes a profound intervention in a landscape. These processes usually transform original habitats and leave them in altered states. However, they do not necessarily lead to a loss of biodiversity (Prach et al. 2011). During and after mining activities, a mosaic of habitats in different successional stages is often created. As surrounding landscapes of mining areas in Central Europe are mostly agriculturally intensively managed, fragmented, and urbanized, such habitat mosaics may act as secondary refuges for several insect groups, such as butterflies, dragonflies, ants, bees, and wasps (Beneš et al. 2003, Ottonetii et al. 2006, Topp et al. 2010, Heneberg et al. 2012, Harabiš and Dolný 2015, Trnka and Rada 2015). However, it is important to use appropriate technical interventions and reclamations, both during mining operations and after quarrying ceases, to achieve beneficial environmental conditions towards the target habitats and/or species alongside the preservation of surrounding environments (Řehounek et al. 2015).

For instance, in many gravel pits of Central Europe, technical reclamations still prevail or are even mandatory, which often lead to a reduction of habitat mosaics created by mining (Heneberg et al. 2016). These processes result in landscape simplification because reclaimed sites are usually covered by soils rich in organic matter and used as an intensive agricultural area (arable fields or hay meadows) or are afforested. Therefore, less competitive, but from conservation point of view valuable assemblages, cannot develop there due to the loss of periodically disturbed habitats that are crucial for their colonization and consequent persistence (Řehounek et al. 2015). Another restoration option is to leave post-mining areas to spontaneous succession or succession with artificial interventions. When spontaneous succession is allowed, it usually leads to secondary forest after approximately two or three decades (Prach et al. 2014). However, spontaneous succession may also be a threat for successful restoration in gravel pits, because these areas are located in floodplains of large rivers (a source of gravel deposits), which are often vectors of invasive alien plants, such as giant goldenrod (*Solidago gigantea*) and giant knotweed (*Reynoutria sachalinensis*). Their invasion may result in dense mono-specific vegetation stands and degradation of native plant, as well as animal, assemblages (Řehounková and Prach 2008, Řehounek et al. 2015).

To correctly assess the ecological value of habitats within gravel pit premises, some bioindicators should be selected. Ground beetles (Coleoptera: Carabidae, hereafter carabids) are often used as good indicators of changes in land-use due to their sensitiveness to various environmental factors both biotic and abiotic. Their ecology and habitat requirements are relatively well-known (Lövei and Sunderland 1996, Altieri 1999, Rainio and Niemelä 2003). In post-mining areas, their assemblages have been intensively studied in brownfields (Eyre et al. 2003, Small et al. 2003), spoil heaps (Schwerk 2004, Hodeček et al. 2015, 2016), limestone quarries (Tropek et al. 2010, Nováková and Šťastná 2013a,b, 2014), and open-cast lignite mines (Brändle et al. 2000). These studies have revealed that the composition of carabid assemblages is determined by the type of reclamation, management, and successional stage of post-mining areas. In gravel pits, carabids were, however, studied only in xerothermophilous grasslands (Heneberg et al. 2016), thus the conservation potential of other habitats is remained unknown.

In this study, we focused on carabid assemblages in a Central European gravel pit surrounded predominantly by intensively managed agricultural landscape in the floodplain area. We supposed that a heterogeneous habitat mosaic created during quarrying could provide suitable conditions for various carabid assemblages. Within the studied gravel pit, we focused on sludge deposits and an agriculturally reclaimed area, sites that could be potentially valuable from a nature conservation and biodiversity perspective. These plots were represented by different stages of succession, management (biannual mowing) and cover of invasive plant species, especially *S. gigantea*, which, as the most dominant alien species in the study area, can create dense mono-specific stands. We addressed the following questions: (i) which habitat (i.e., plot) supports the occurrence of species-rich, abundant and diverse carabid assemblages or red-listed species? (ii) are carabids affected by management practices and (iii) does the cover of *S. gigantea* have negative impact on carabids assemblage?

Material and methods

Study area

The studied gravel pit is located near the town Hulín in the eastern part of the Czech Republic (GPS: 49.299N, 17.455E, 191 m a.s.l.). The study plots were around a gravel mining lake including sludge deposits in the eastern part and agriculturally reclaimed area in the southern part of the gravel pit premises. Sludge deposits, which are repositories of fine sand particles as remains of gravel washing that gradually dried up, were abandoned for spontaneous succession after the termination of infilling processes. They represent a habitat mosaic in different successional stages from bare sandy soils to secondary forest. The reclaimed area was originally founded as extensively managed grassland but some parts were left without regular management and now mostly covered by invasive giant goldenrod (*S. gigantea*).

We focused on four sludge deposits (hereafter as plots SD1–SD4, each roughly 2 ha, Fig. 1), each with a differently developed vegetation cover due to different successional stages. SD1 (bare sand): three years old sludge deposit with sandy substrate that still remains barren without any vegetation cover, only the wetter marginal areas were overgrown with reed (*Phragmites australis*). SD2 (reed stands): 18 years old uniform cover of reed (*P. australis*) with birches (*Betula pendula*) at the border of this sludge deposit. Under birches, vegetation was represented by various grasses without bare ground. SD3 (young forest): 20 years old birch (*B. pendula*) forest with sandy substrate without high vegetation cover of the herbal layer but with leaf litter. Moreover, a part of the area was disturbed by movements of heavy machinery in 2016, thus new bare sandy patches without any vegetation cover existed here as well. SD4 (old forest): 30 years old secondary forest dominated by willows (*Salix* sp.), poplars (*Populus* sp.), and birches (*B. pendula*) and by *Urtica dioica* and *Galium aparine* in the herb layer.

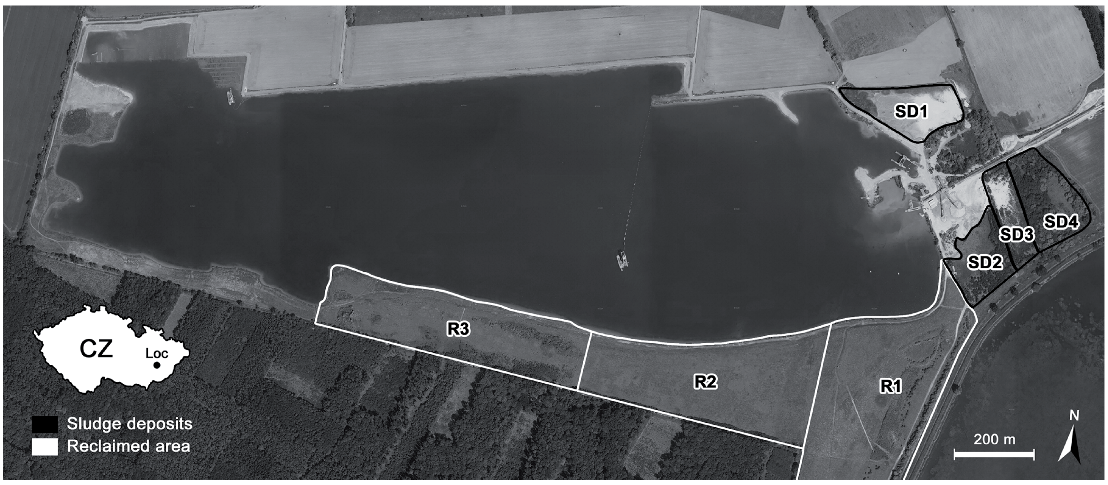

Figure 1. Map of the Hulín gravel pit with the studied plots. SD plots are sludge deposits, R plots correspond with the agriculturally reclaimed area.

Table 1. Characteristics of studied plots with coding of environmental variables.

Plot	Area	Habitat	Management	Cover of *Solidago gigantea* (range)
SD1	Sludge deposits	Bare sand	no	0
SD2	Sludge deposits	Reed stands	no	0
SD3	Sludge deposits	Young forest	no	0
SD4	Sludge deposits	Old forest	no	0
R1	Reclaimed area	Grassland	yes	0
R2	Reclaimed area	Grassland with *Solidago* stands	yes	0–0.65
R3	Reclaimed area	*Solidago* stands	no	0.4–0.9

Besides sludge deposits, our second area of interest was the agriculturally reclaimed area, where we studied three plots (R1–R3, Fig. 1) each with different management and cover of giant goldenrod (*S. gigantea*). R1 (grassland, 10 ha): grassland with wet depressions and without any *S. gigantea* cover. It has been sown with a grass seed mixture (20% *Lolium perenne*, 23% *Phleum pratense*, 20% *Festuca pratensis*, 20% *Alopecurus pratensis* and 12% *Poa pratensis*) with a deep cultivation and cutting twice a year. R2 (grassland with *Solidago* stands, 8 ha): grassland with wet depressions that has been sown with the same grass seed mixture as in the plot R1 but with a shallow cultivation. Due to insufficient plowing preceding the sowing, plot R2 was covered by species-poor stand where *S. gigantea* predominate. The plot was also mown twice a year. R3 (*Solidago* stands, 9 ha): after sowing with grass seed mixture, this plot was left to spontaneous succession for 20 years and recently its majority was overgrown by *S. gigantea* with small groups of willows (*Salix* sp.) and poplars (*Populus* sp.) in the plot center.

Carabid sampling

The composition of carabid assemblages was recorded in sludge deposits and the reclaimed area. Individuals were collected using pitfall traps consisting of two plastic cups filled with 4% formaldehyde and checked every three weeks (i.e., sampling session) during the main period of carabid activity from May to September 2016. In total, six sampling sessions have been conducted. In sludge deposits, we placed nine traps into each plot (i.e., SD1–SD4) to cover various microhabitats. Since plots in the reclaimed area were more uniform in terms of vegetation structure than in sludge deposits, only four traps were installed in each plot (i.e., R1–R3). Traps were placed in the plot center, 10 meters from each other in a square of 3 × 3 traps (sludge deposits) or in a transect of four traps (reclaimed area). In total, we installed 48 traps and recorded the type of area (sludge deposit or reclaimed area), habitat, which represented overall vegetation structure and cover in the particular plot, management and the cover of *S. gigantea* within a two-meter radius around each trap (Table 1 provides the coding of environmental variables). Collected material has been sorted in the laboratory and carabids were identified to the species level following Hůrka (1996). Moreover, we have assigned a conservation status to each of the recorded species according to the Czech Red List of Invertebrates (Veselý et al. 2017). For further analyses, we calculated the number of species and individuals in each trap. No samples pooling was required because the time period between sampling sessions had the same length in all cases.

Statistical analyses

To measure the effectiveness of sampling, rarefaction analyses were conducted for each plot. The species richness was estimated with the Chao estimator (Chao 1987) using the *specaccum* function from the 'vegan' package (Oksanen et al. 2019). The standard deviations were generated from 10,000 reshufflings of the sample order. Non-metric multidimensional scaling (NMDS) was applied to test dissimilarities in the composition of carabid species between studied plots based on the Bray-Curtis similarity matrix for abundance data, using the *metaMDS* function from the 'vegan' package. The diversity of carabids in different habitats was compared by Rényi's one-parametric diversity index family using the *renyi* function from the 'vegan' package. This approach enables visualization of diversity profile of the assemblage by plotting several diversities against a scale parameter (Tóthmérész 1995, Lövei 2005, Ricotta 2005). When the scale parameter is zero, the plotted value represents the logarithm of species richness. In this stage, the index is sensitive to abundance of rare species. The Rényi diversity corresponds to the Shannon diversity when the scale parameter reaches one. When the scale parameter is two, the plotted value is the quadratic (Simpson) diversity. Further increase of the scale parameter towards positive infinity results in values related to the Berger-Parker dominance index and it is sensitive to presence of common and abundant species.

For testing responses of carabid species richness and abundance to environmental and management variables, we used generalized linear mixed models (GLMMs) with Poisson distribution. Firstly, we created two sets (one for species richness, one for abundance) of single-argument models with only one explanatory variable included to avoid collinearity (Burnham and Anderson 2002). As explanatory variables, type of area (categorical variable, two levels), plot (categorical, seven levels), management (categorical,

Table 2. Summary of the model selection; using estimations based on the calculated AICc value of the models, serving as the weight of evidence in favor of the different models. Most parsimonious models (delta < 2) are emphasized in bold.

Response	models	df	logLik	AICc	delta	weight
Species richness	habitat	**8**	**-366.627**	**750.0**	**0.00**	**0.995**
	area	3	-377.471	761.1	11.03	0.004
	management	3	-379.296	764.7	14.67	0.001
	cover of *Solidago*	3	-381.996	770.1	20.08	< 0.001
Abundance	habitat	**9**	**-1105.980**	**2228.8**	**0.00**	**> 0.999**
	area	4	-1167.689	2341.5	112.76	< 0.001
	management	4	-1175.648	2357.4	128.67	< 0.001
	cover of *Solidago*	4	-1191.252	2388.6	159.88	< 0.001

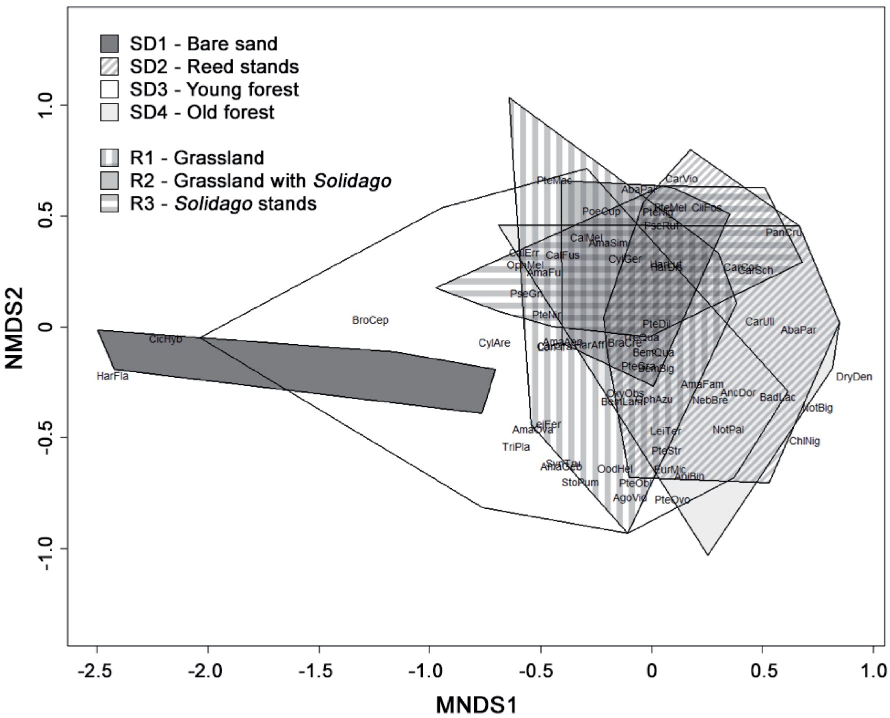

Figure 2. Non-metric multidimensional scaling ordination of the total abundance of carabids captured in different plots using the Bray-Curtis similarity matrix. Species names are abbreviated with the first three letters of genus and species name; see Table A1 for full names.

two levels) and cover of *S. gigantea* (continuous) were used in the *glmer* function from the 'lme4' package (Bates et al. 2015). Sampling period was considered as a random effect in the models. Then, using *model.sel* function from 'MuMIn' package (Bartoń 2019), each model set was tested to select the best model(s) from given set based on information criterion corrected for small samples sizes (Akaike Information Criterion – AICc, Burnham and Anderson 2002). The best model was selected as the most parsimonious explanation of the data, when Δ AICc was higher than two for other models from the given set (Table 2). If any significant differences for explanatory variables were detected, we used post-hoc test with Tukey contrasts for multiple comparisons of means (Bretz et al. 2010) by the *glht* function from the 'multcomp' package (Hothorn et al. 2008). All analyses were conducted in R 3.4.1 (R Core Team 2017).

Results

Assemblage characteristics and diversity

A total of 64 carabid species and 1920 individuals were identified. The species richness estimates revealed continuous increase in all plots as the number of samples increased (Fig. A1) suggesting some missing species. The list of species recorded and their distribution among the plots studied are provided in Table A1. Three species are present in the Czech Red List of Invertebrates as vulnerable: *Pterostichus gracilis*, or near threatened: *Cylindera germanica*, and *Harpalus flavescens*.

NMDS (stress value = 0.123, Fig. 2) showed a gradient in carabid assemblages, from psammophilous ones in bare sandy soils (plot SD1 and partially also SD3) towards assemblages occurring in plots with well-developed vegeta-

Table 3. Summary of the best models; significant effects are in bold.

Response	best models	variables	χ2	df	p
Species richness	Habitat	(intercept)	**99.144**	1	**< 0.001**
		habitat	**28.020**	6	**< 0.001**
Abundance	Habitat	(intercept)	**114.180**	1	**< 0.001**
		habitat	**150.76**	6	**< 0.001**

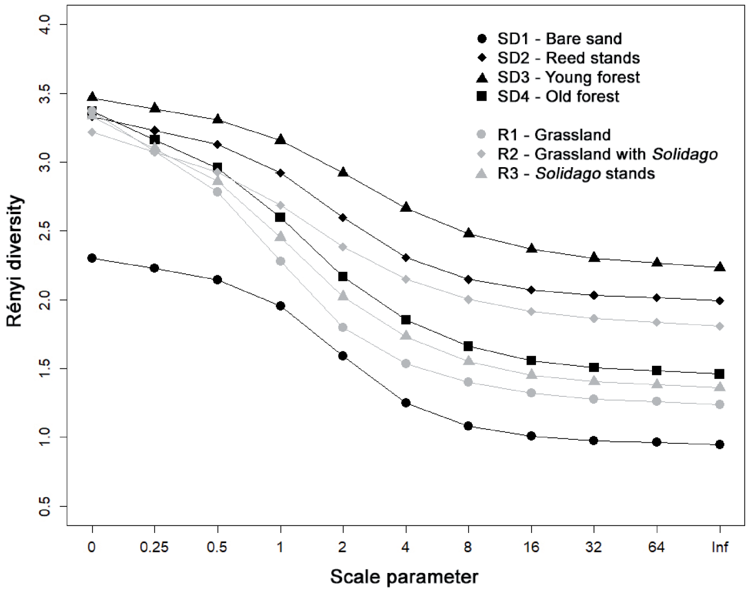

Figure 3. Diversity profiles of the carabid assemblages of studied plots by the Rényi one-parametric diversity index family.

tion cover. Here according to NMDS, carabid assemblages were similar regardless of plot type. Plot SD1 had the lowest Rényi diversity (Fig. 3) with species of dry habitats, namely *Cicindela hybrida*, as the most abundant and red-listed *H. flavescens.* Contrary, the highest diversity was in SD3 due to high inter-plot heterogeneity ranging from bare soils to young birch forest. Typical species of the disturbed sandy patch were xerothermic and less common species (but not recently red-listed) *Broscus cephalotes* and *Cylindera arenaria*. The assemblage in the secondary old forest (SD4) was characterized by large species typical for seminatural forests, dominated by *Carabus scheidleri* and *Abax parallelepipedus*. The species composition in the reclaimed area was similar for all (R1–R3) plots. The Rényi diversity was the highest in grasslands with *Solidago* stands (R2) and the lowest in grasslands (R1) without any cover of this invasive plant. In contrast to three red-listed species (we also found *P. gracilis* in reed stand SD2 and *C. germanica* in sandy patch in the young forest SD3) and several habitat specialists recorded in sludge deposits, only the extensively managed grassland R1 supported the occurrence of near threatened species of fallow lands and pastures (*C. germanica*) in the reclaimed area.

Responses to management and environmental factors

The model selection based on the calculated AICc revealed that the 'habitat' model was the most parsimonious explanation of the data for both carabid species richness and abundance (Tables 2 and 3). We found no effect of management practices and the cover of *S. gigantea* on carabids. In sludge deposits, both taxonomical indices were the lowest in plot SD1 (Fig. 4a and 4b). Whereas species richness did not differ between other sludge deposits, abundance was higher in young forest SD3 than in other plots. In the reclaimed area, there were no significant differences in species richness between plots, however, abundances differ considerably revealing the same values in grassland (R1) and *Solidago* stands (R3) and the lowest in grassland with *S. gigantea* (R2).

Discussion

Our findings showed a gradient in carabid assemblages ordered according to the presence of bare sandy soils and vegetation cover at the ground level. These changes in assemblage composition, numbers of species and individuals are associated with the development of spontaneous succession (Schwerk 2004, Magura et al. 2006, Hodeček et al. 2015, Kašák et al. 2017, Šipoš et al. 2017). Carabid diversity was also related to inter-plot habitat heterogeneity because the highest values were observed in the young birch forest with both bare sandy soils and leaf litter under trees. Since sludge deposits were relatively small in size (approximately two hectares each) apparently the spillover and edge effect also played an important role in carabid distribution within and between plots (e.g., Magura 2002, Tscharntke et al. 2005, Elek and Tóthmérész 2010, Magura et al. 2017).

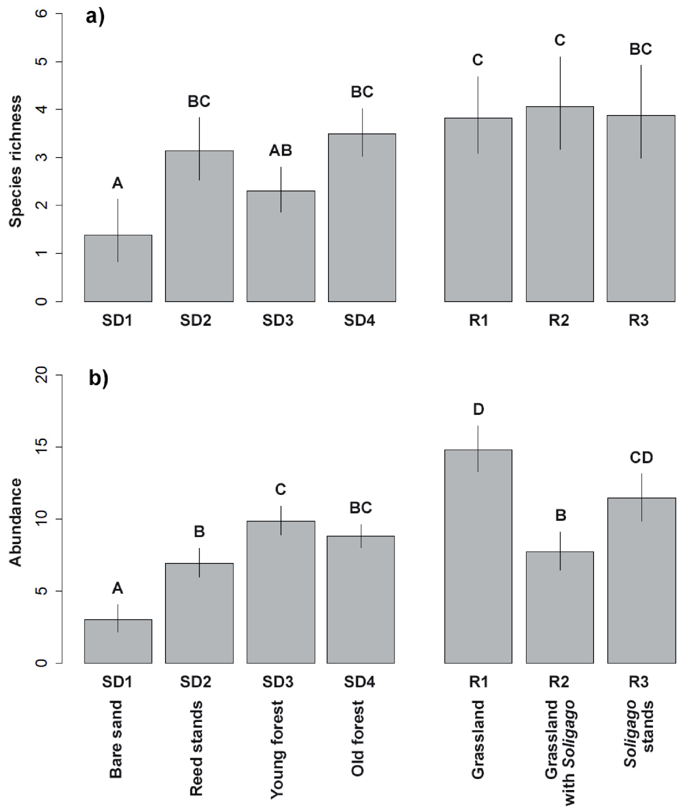

Figure 4. Mean species richness (**a**) and abundance (**b**) of carabids per trap and sampling season in different habitats of sludge deposits (SD plots) and the reclaimed area (R plots). Error lines represent 95% confidence interval and capital letters above indicate significant differences based on post-hoc Tukey's pairwise multiple comparisons.

For instance in the secondary forest, although the assemblage was represented mostly by forest species, such as *Carabus* spp. and *Abax* spp. (see Table A1), typical species of arable fields (*Anchomenus dorsalis*, *Poecilus cupreus*, see Table A1) that are very common in surrounding agricultural landscape (Honěk 1997, Veselý and Šarapatka 2008) were found here as well. Similarly, some typical forest species, such as *Carabus coriaceus*, likely dispersed from adjacent forested areas because they were recorded in the reclaimed area. The secondary forest, reed stands and reclaimed plots had the most species-rich carabid assemblages within the gravel pit but regarding the conservation value, the number of red-listed species was not considerably higher than in other plots.

Contrary, habitats in early successional stages support compositionally different assemblages including open habitat specialists, such as the psammophilous *B. cephalotes*, *C. arenaria* or *H. flavescens*. As natural open sands are very rare in the urbanized, fragmented, and intensively managed agricultural landscape of Central Europe, the occurrence of psammophilous species is often limited to a few suitable sites along exposed shores of unregulated parts of large rivers (Riksen et al. 2006, Fanta and Siepel 2010, Tropek and Řehounek 2011). Gravel pits with newly emerging sandy habitats with an exposed substrate as a consequence of quarrying processes therefore have a potential to supports these species of carabids, as well as other rare insects (Heneberg et al. 2016). This role of gravel pits can also be emphasized by the record of earwig *Labidura riparia*, red-listed as near threatened in the Czech Republic, in sandy patches of SD3. We presume that these species cannot persist outside quarried premises. Nevertheless, such open habitats need regular disturbances otherwise they will overgrow due to continued succession and their potential as a refuge for psammophilous species will be devalued (Schwerk 2004). The effective management should be aimed at maintaining habitat mosaics with an emphasis on open sands, sandy walls, and early successional stages of vegetation (Řehounek et al. 2015). Required disturbances can be mostly formed by normal operations in the gravel pit, namely by movements of heavy machinery and/or of material from sludge deposits. It is important to note that our findings are based on a relative small sampling without spatial replication. Making general conclusions and recommendations is thus limited. Nevertheless, we believe that this case study is a nice example how early successional habitats can be important for dry habitat specialists and enhance their persistence in intensively managed agricultural landscapes.

We found no carabid response to management. Likely, changes in vegetation structure in plots with extensive management are not so high to significantly affect the carabid assemblage. When vegetation cover (especially at the ground level) is fully developed, for carabids, it does not matter if the plot is extensively managed or left to spontaneous succession. Carabids respond more to vegetation structure (density, proportion of bare soils) than to species-specific composition of plant assemblages (de Groot et al. 2007, Humbert et al. 2009). They are surface-dwelling arthropods and use vegetation mainly for foraging and as a shelter (Eyre et al. 2003). This can be a reason why we did not find a significant negative effect of *S. gigantea* cover on carabid diversity or activity density. *Solidago* stands can be very dense sometimes (Řehounková and Prach 2008) and therefore can potentially slow down movement activity of the entire carabid assemblage or only some species (Mauremooto et al. 1995, Baranová et al. 2014, Ranjha and Irmler 2014, Růžičková and Veselý 2018). This limitation can explain why previous studies have reported a reduction in diversity (Baranová et al. 2014) and lower abundances (de Groot et al. 2007) of carabids in *Solidago*-invaded plots. The significant effect (negative, as well as positive) of invasive plants is particularly profound for species that rely heavily on plants as essential resources of food, such as herbivores and pollinators, which make them susceptible to the intrusion of invasive plants (Ernst and Cappuccino 2005, de Groot et al. 2007, Baranová et al. 2014).

Conclusion

In gravel pits, habitats in various successional stages, including bare sands, are often removed during technical reclamations; covered by topsoils and secondarily afforested or used as an agricultural area. These reclamations are mandatory in some EU countries, including the Czech Republic (Heneberg et al. 2016). In our case, future technical reclamations of sludge deposits mean that the already existing habitat mosaic will be converted to a simplified landscape similar to surrounded agricultural areas. For rare psammophilous carabid species that almost disappeared from the agricultural landscape of Central Europe due to habitat loss, such drastic changes in environmental conditions may be detrimental for their occurrence in the gravel pit. Maintaining bare sandy patches acting as refuges, stepping-stones, and corridors are therefore highly recommended for their persistence. Although our sampling concerns at the local-scale, we demonstrate that some highly specialized species (e.g., *H. flavescens*) will able to colonize these new suitable habitat patches. It may guarantee the success of ecological restorations of habitat patches beyond the gravel pit premises, such as river banks, where these species originally come from.

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Appendix

Figure A1. Estimations of carabid species richness according to Chao estimators in sludge deposits and reclaimed area.

Table A1. A list of recorded carabids and their occurrence in studied plots.

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